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LAND USE IN LCA

A comparison of three methods to assess land use impacts on biodiversity in a case study of forestry plantations in New Zealand

Ottar Michelsen · James E. McDevitt · Carla R. V. Coelho

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Abstract
Purpose The aim of this study is apply available methods to assess impacts on biodiversity from the land use caused by plantation grown radiata pine in New Zealand in a life cycle assessment framework. This is done both to quantify the impact as well as compare the results obtained by different methods.
Methods Data on location and productivity for wood supply regions in New Zealand was assessed using three methods identified as relevant for the purpose. All data were related to a functional unit of 1 m^3 of timber production.
Results and discussion The results show both a significant difference in impact on biodiversity from land use in the different wood supply regions and a significant difference in the results from the three applied methods. Although some of the results obtained from the three methods were correlated, this was not consistent through all the results. The methodological variation emanates from the treatment of the characteristics of the wood supply regions and underlying assumptions, e.g. reference vegetation. Compared to a case study in Norway, the impact on biodiversity from plantation forestry in New Zealand is found to be relatively low following the applied methods and assumptions taken.
Conclusions The study shows a significant variation in how impacts on biodiversity are assessed following different approaches. Research to harmonize methods to quantify impact on biodiversity is recommended, as well as exploring effects of different weighting of crucial aspects of biodiversity, such as rarity, abundance and species richness.

Keywords Biodiversity · Life cycle assessment · New Zealand · *Pinus radiata* · Plantation forestry

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O. Michelsen (✉)
The Industrial Ecology Programme, Norwegian University of Science and Technology (NTNU), 7491 Trondheim, Norway
e-mail: ottar.michelsen@ntnu.no

J. E. McDevitt
Scion, Te Papa Tipu Innovation Park, 49 Sala Street, Rotorua 3010, New Zealand

C. R. V. Coelho
Geospatial Programme Delivery, Auckland Council, L7 8 Hereford Street, Auckland 1011, New Zealand

1 Introduction

There is an increasing demand for more knowledge on the impact from products and services provided in the global market. Consumer influence in supply chain management is a nascent market force being realized (Michelsen et al. 2006). Product-related claims pertaining to sustainable resource management in production systems such as agriculture, forestry and mining are a particular focus for the equitable use of resources. One of the critical issues for debate is impact on biodiversity on a global scale (Lenzen et al. 2012) as biodiversity is an immeasurably precious resource (Gómez-Baggethun and Ruiz-Pérez 2011; Tscharntke et al. 2012).

Life cycle assessment (LCA) is a comprehensive method for assessing impacts caused by products, services and systems. One primary driver behind the use of LCA is that it is regarded a transparent and verifiable method to quantify impact. There are however still shortcomings (Finnveden et al. 2009) and in particular impacts on biodiversity is only partly included (Curran et al. 2011).

Biodiversity can be defined as "the variability among living organisms from all sources […] this includes diversity within species, between species and ecosystem" (United Nations 1992) and the value of biodiversity is well recognized as an essential life support function (Milà i Canals et al.

2007b). Loss of biodiversity is an important impact of anthropogenic activities and one of the most severe threats to sustainability (Diaz and Cabido 2001; Butchart et al. 2010), but the complexity of biodiversity including different taxonomic levels, biological attributes and temporal and spatial dimensions makes it difficult to assess changes over the full range. Consequently, assessments of biodiversity have to simplify this complexity and associate it to the current state of knowledge; for example via focusing on one region (e.g. Kohlmaier et al. 2007) or taxa (e.g. Landeiro et al. 2012). This complexity is also challenging when impacts on biodiversity is to be included in LCA.

For the time being, land use and land use modifications are considered being the main drivers for loss of biodiversity (Chapin et al. 2000; Sala et al. 2000; Henry et al. 2008; Haines-Young 2009; Lenzen et al. 2009). There is an increasing number of proposals on how impacts from land use on biodiversity could be included in a LCA framework (Köllner 2000; Kohlmaier et al. 2007; Michelsen 2008; Geyer et al. 2010; de Baan et al. 2013a, b; Coelho and Michelsen 2014), but so far no consensus on how to assess this has been reached (Koellner et al. 2013b). Land use impacts on biodiversity can be further subdivided into three types of impact: transformational impacts, impacts from occupation and permanent impacts, e.g. species extinction, which needs to be treated differently due to their irreversible character. Most proposals for occupational impact are based on the premise that land use impacts should be assessed in three dimensions: area, quality and time where the basic idea is that a stretch of land (area) is altered from its current state (quality) to adapt it to the intended use for a period of time (Milà i Canals et al. 2007a; Koellner et al. 2013b). While the area of a certain impact is relatively easily determined, quality changes and the time scale of the alteration are much harder to assess. The calculation of transformation impacts from land use requires reliable data on regeneration success and timeframes, and there is a significant paucity of reliable data for this. The reference state from which the quality difference is to be related to is also questioned (e.g. Milà i Canals et al. 2007a) where the present situation as well as a natural or potential natural vegetation (PNV) being the most relevant (Koellner et al. 2013b). Different methodological approaches might give rise to different conclusions and thus make results questionable for decision support (Anton et al. 2007).

The aim of this study is to apply available methods to assess impacts on biodiversity from the land use caused by plantation-grown radiata pine (*Pinus radiata* L.) in New Zealand. Data on location and productivity in different wood supply regions (WSR) will be used with different methods for assessment of land use impact on biodiversity. This will be used to evaluate both differences in impacts in the WSRs as well as differences in the available methods. Two different reference situations are also used in order to evaluate the differences in choice of reference.

The forest industry in New Zealand is based around managed exotic forest plantations, covering 1.8 million hectares, approximately 9 % of New Zealand's land area (MfE 2013b). New Zealand supplies almost 9 % of the Asia Pacific forest products trade volume, representing 20 % by value (MAF 2012), being accountable for approximately 3 % of New Zealand's export earnings (MAF 2012). Ninety percent of the exotic plantation area comprises radiata pine, with douglas fir (*Pseudotsuga menziesii* L.) accounting for 6 % and the remainder of the estate made up of eucalyptus and other softwood and hardwood species (MAF 2010). Radiata pine is an exotic tree species for New Zealand and is indigenous to California at Ano Nuevo, Monterey and Cambria and two island populations in Guadalupe Island and Cedros Island (Shelbourne et al. 1979). It was introduced to New Zealand in 1859 and was first grown in Mt Peel Station in South Canterbury (Berg 2009). New Zealand's land cover consists of 50 % native vegetation (indigenous forest and other native land cover) and nearly 40 % pasture (MfE 2013b).

2 Methods

2.1 Goal and scope

The goal of this study was to use appropriate methods to calculate an estimate of the impact on biodiversity due to land occupation of forest plantations in the WSRs of New Zealand. The functional unit for each region is 1 m^3 of harvested wood.

2.2 Forestry production data

The growth rates for the different WSRs are adapted from MAF (2010), adjusting for the different management regimes based on the area that is subject to that management regime. The area under forest, the growth rate and average harvest cycle for the different WSRs are presented in Table 1. Further details on the modeled forestry production are given in outlined in Nebel and Drysdale (2009).

2.3 Reference situation

Two potential reference situations are selected: native vegetation and pastures. Native vegetation is here regarded as equal to PNV which often is used as reference in land use impacts in LCA (Koellner et al. 2013b). In New Zealand, pastures, covering nearly 40 % of the country, are in most cases a more realistic place for establishment for plantations and are included as an alternative reference situation to show differences in the use of PNV versus present vegetation as reference for land occupation impact.

Table 1 Growth rate and the area under plantation *P. radiata* for included wood supply regions from the National Exotic Forest Descriptions 1990–2013

Region	Average growth rate (m^3/ha/year)	Area under forest (ha)	Average time between planting and harvest (years)
Northland	18.2	202,577	29.1
Central North Island (CNI)	21.6	551,628	28.8
East Coast	17.5	154,289	29.6
Hawkes Bay	17.6	129,563	30.0
Southern North Island (SNI)	18.6	165,622	29.7
Nelson and Marlborough	19.1	168,373	28.5
West Coast	11.0	32,466	29.2
Canterbury	17.3	110,032	31.0
Otago and Southland	16.4	203,729	32.9

2.4 Methodologies for assessing impacts on biodiversity

Most methods where land use impacts are included in LCA are based on the assumption that an area is altered from its current state (quality) to adapt it to the intended use for a period of time and total impact should be assessed by integrating over area, time and quality changes (Milà i Canals et al. 2007a; Koellner et al. 2013a). In particular, assessment of quality changes and the time scale of the alteration are debated and at the time being there is no consensus on how to include this in LCA. Figure 1 shows the basic idea behind this and also consequences of different reference situations. Here an activity, e.g. forestry, is assumed to take place between t_0 and t_{fin}, where t_{fin} represents the end of that activity. In the case of forestry, this period will be divided into a number of rotations. The impact is the time and area needed for producing a functional unit (in this case study 1 m^3 of timber), and the quality differences this causes for the area. This is shown by the grey area in Fig. 1. Different reference situations can be used. If forestry causes a quality Q_j in the figure, the impact will in most cases be assessed higher if PNV is used (Q_{pot}) compared to if present use (e.g. agriculture) is used as reference, in the figure shown as Q_i. There will of course also be situations when the quality increases due to a land activity, e.g. if the quality is changed from Q_j to Q_i due to changes in land use. If present land use is used as reference, this will be assessed as a quality improvement (from Q_j to Q_i), but if PNV is used as reference, this will still be a negative impact (change from Q_{pot} to Q_i). See also Milà i Canals et al. (2007a) for more discussion on reference situations.

In this paper, only occupational impact from the consecutive rotations is taken into account and transformation impact caused by the first land use change, shown as hatched area in the figure, is not included. With an increasing number of

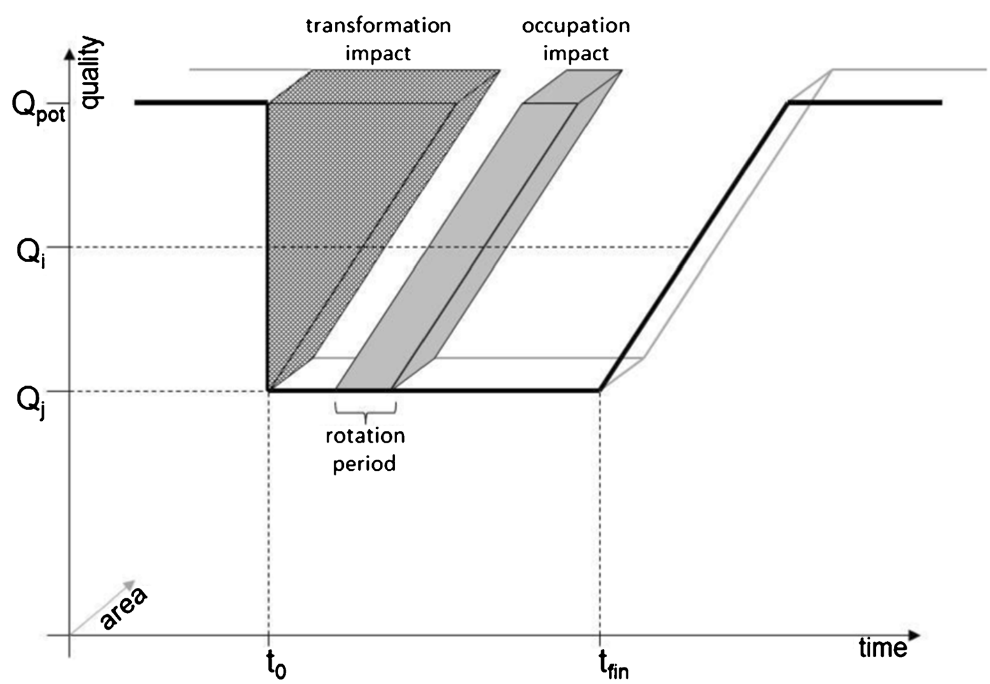

Fig. 1 Land occupation impact is assessed as a product of time, area and quality changes. Here an ongoing activity is assumed to take place from t_0 to t_{fin}, but a rotation period lasts only part of this. Total quality is depending on reference situation, e.g. if Q_j is related to Q_{pot} or Q_i. See text for more details

rotations, this will make a minor difference (Michelsen et al. 2012).

Suggested methods on how to assess quality and changes in quality in terms of biodiversity can at first be divided into two main categories focusing on (1) changes in species composition and (2) structural changes: habitats, key factors, etc. (Michelsen et al., in preparation). For all methods on quality assessment, lack of global data has been a problem (e.g. de Baan et al. 2013a), and only three methods are found useful for forest plantations in New Zealand: an approach suggested by Michelsen (2008) and further adjusted by Coelho and Michelsen (2014) focusing on ecosystem scarcity and vulnerability, and two methods proposed by de Baan et al. (2013a, b) focusing on relative changes in species richness and absolute loss of species, respectively. Other potential approaches are discussed later in the paper.

Michelsen (2008) suggested to assess the biodiversity quality (Q) and changes in quality at a given location and time, indirectly based on ecosystem scarcity (ES), ecosystem vulnerability (EV) and conditions for maintained biodiversity (CMB), using the formula

$$Q_{\mathrm{t}} = \mathrm{ES}_{\mathrm{t}} \times \mathrm{EV}_{\mathrm{t}} \times \mathrm{CMB}_{\mathrm{t}}. \tag{1}$$

The rationale behind the proposal is that ecosystem rarity is proportional to the value of the ecosystem. Due to global data availability, Michelsen (2008) suggested to use ecoregions as spatial resolution, dividing the earth into 867 ecoregions (Olson et al. 2001). Ecosystem scarcity is a measure of the intrinsic rareness of an ecosystem while ecosystem vulnerability is a measure on the present condition of the ecoregion. Michelsen (2008) used a three graded scale given in WWF's Wild Finder (2013), while Coelho and Michelsen (2014) suggest an equation based on fraction still left of original vegetation.

CMB is an index for the actual impact on the biodiversity in the affected area, ranging from 1 which represents no impact on biodiversity to 0 which represents a complete removal of biodiversity. The CMB index is constructed by a suite of indicators known to be important for biodiversity in the ecosystem and will thus give an indirect measure on the impact on biodiversity (cf. Larsson 2001). Michelsen (2008) constructed a CMB index for boreal forests, but since such indicators are not transferrable to other ecosystems and ecosystem-specific indicators are lacking for most ecosystems, Coelho and Michelsen (2014) suggest using values on level of naturalness from Brentrup et al. (2002) as a proxy. In the case presented in this paper, hemeroby class H4 is used for forest plantations, giving a CMB at 0.6, while hemeroby class H5 is used for pastures (medium intensity).

These three parameters combined in accordance to Eq. 1 is used to determine the quality (Q) of the land at a given point in time (t) and quality changes between two points in time can consequently be calculated. It is noteworthy that all these parameters yield data between 0 and 1 and consequently Q is between 0 and 1. Data requirement for this approach is productivity (m^2 year/functional unit) and location to ecoregion.

Most methods for assessing quality changes in terms of biodiversity is based on species richness and in most cases species richness is related to a regional reference, giving relative values. This is also the starting point for the proposal by de Baan et al. (2013a). Their proposal is the most recent of the approaches using relative changes in species richness and is also the first with a global approach. Most of the previous methods, such as the species-pools effect potentials method by Köllner (2000) and further developments of this, are based on limited datasets, primarily from Europe, and are not regarded valid for New Zealand. de Baan et al. (2013a) develop characterization factors for land use activities on biome level. The regional reference situation for species richness is a current late-succession habitat stage which areas under human influence are assumed to revert to if the present use is abandoned.

Even with a rough division into eight land use types and 14 biomes, there is still lack of data for parts of the world. The relevant biomes for New Zealand are 'Temperate broadleaf and mixed forests', 'Temperate grasslands, savannas, and shrublands' and 'Montane grasslands and shrublands'. Plantations are covered by the land use class 'Forest, used' while pastures are represented by the land use class 'Pasture/meadow'. Since de Baan et al. (2013a) do not provide a characterization factor for 'Forest, used' for 'Temperate grasslands, savannas, and shrublands', a world average for 'Forest, used' is used as a starting point. To check the sensitivity of this choice, we also include worst case value for the CF, in this case the highest CF for 'Forest, used' registered worldwide. Data requirement for this approach is productivity (m^2 year/functional unit) and location to biome.

de Baan et al. (2013a) uses PNV as a reference situation for their characterization factors. This means that plantations established in replacement for native vegetation or on pastures have the same impact (cf. Fig. 1). Jørgensen et al. (2013) questioned this and suggested a modification where the impact is calculated as total impact of actual situation minus the total impact of the present situation, allowing also for a negative impact if the land use in focus represents an improvement of biodiversity compared to the previous land use.

de Baan et al. (2013b) have developed a method for occupational impact based on absolute loss of non-endemic species. Loss of endemic species is not included in occupational impact since they regard this as a permanent impact and treat this separately. de Baan et al. (2013b) provide characterization factors for four land use classes: agriculture, pasture, urban areas and managed forest for five taxonomic groups (mammals, birds, plants, amphibians and reptiles) as well as an

aggregated value on ecoregion level. This is a method with a much higher spatial resolution that other species based proposals, but with a low number of land use classes. For occupational impact from forest plantation, characterization factors based on aggregated values for all included taxonomic groups for managed forest is used. Data required for this approach is productivity (m^2 year/functional unit) and location to ecoregion. Similarly as in de Baan et al. (2013a), PNV is used as reference situation, but pastures are used as an alternative reference situation also here.

A challenge with all these methods is that some of the identified WSRs (Table 1) are located to multiple ecoregions and biomes. New characterization factors and data on ecosystem vulnerability and ecosystem scarcity is thus computed as a weighted average, following the equation

$$\mathrm{CF}_r = \sum_{i=1}^{n} (\mathrm{CF}_i \times \mathrm{fraction}_{ir}) \tag{2}$$

where CF_r is the characterization factor used for the WSR r, CF_i is the characterization factor for the relevant biome or ecoregion i and $\mathrm{fraction}_{ir}$ is the fraction of the biome or ecoregion i in WSR r given by the relative distribution of the plantations on each ecoregion for the different WSRs. This is assessed using GIS to identify the location of the plantations which is related to WSR and ecoregions (Fig. 2). The spatial data on WSRs and ecoregions (WWF 2013) were overlaid using GIS. For the spatial distribution of the plantation forest, we used Land Use Map which resulted from New Zealand's Land Use and Carbon Analysis System (MfE 2013a), which is publically available and free of charge at www.koordinates.com. The amount of plantation area for each WSR and ecoregion were calculated using GIS. The calculation of new scores for ES and EV follows the same line. Necessary values for assessing land use impacts for the different WSRs are shown in Table 2.

Fig. 2 Overview over the distribution of wood supply regions, ecoregions and plantation forestry in New Zealand

3 Results

The impacts for land occupation for plantation-grown *P. radiata* over the different regions in New Zealand using the selected approches with the two different reference situations are shown in Figs. 3 and 4 and Table 3. All values are a combination of a quality measure multiplied by area and years to provide 1 m^3 of timber.

Since de Baan et al. (2013a) do not provide a CF for 'Forest, used' for the biome 'Temperate grasslands, savannas, shrublands', different choices can be made. In Table 2, a world average equal to 0.18 is used, giving the results in Figs. 3 and 4 and Table 3. The uncertainty of this is high and to test the sensitivity also a worst case using 0.55 (the CF for the biome 'Montane grasslands and shrublands') is calculated. This significantly changes the results for the WSRs Canterbury and Otago and Southland where the weighted CF increases to 0.55 and 0.44, respectively, and the total impact to 0.032 and 0.026 (Fig. 4) when PNV is used as reference. When pasture is used as reference vegetation, the total impact for Canterbury and Otago and Southland both changes from a small negative impact (i.e. improved environmental performance) on −0.002 and −0.008, respectively, to an impact on 0.018 and 0.005 (Fig. 4). For all other areas, the changes are insignificant.

Figure 3 reveals a pattern in the results obtained with the three different methods, even though there are clear discrepancies. The results obtained with the method from Coelho and Michelsen (2014) are significantly correlated to both methods from de Baan et al. (2013a, b), but the two methods from de Baan et al. are not correlated (Table 4). If a worst case is used as CF for 'Forest, used' for the biome 'Temperate grasslands, savannas, shrublands', there is no correlation between the results following de Baan et al. (2013a) and Coelho and Michelsen (2014).

Table 2 The data used to calculate the characterization factors for the different regions investigated

Wood supply region	Ecoregion[a]	Biome[b]	CF_{occ_b}[c]	CF_{occ_b}[c, d]	CF_{occ_e}[e]	CF_{occ_e}[d, e]	ES[f]	EV[f]	ΔQ[e]	ΔQ^*[e]
Northland	AA0406	TBF	0.22	−0.30	$6.2e^{-12}$	$-2.2e^{-10}$	0.994	1	0.397	−0.099
Central North Island	AA0405, AA0406	TBF	0.22	−0.30	$4.3e^{-12}$	$-8.3e^{-11}$	0.983	1	0.393	−0.098
East Coast	AA0405	TBF	0.22	−0.30	$4.1e^{-12}$	$-6.5e^{-11}$	0.982	1	0.393	−0.098
Hawkes Bay	AA0405	TBF	0.22	−0.30	$4.1e^{-12}$	$-6.5e^{-11}$	0.982	1	0.393	−0.098
Southern North Island	AA0405	TBF	0.22	−0.30	$4.1e^{-12}$	$-6.5e^{-11}$	0.982	1	0.393	−0.098
Nelson and Marlborough	AA0404, AA0406, AA0408, AA0801, AA1003	TBF, TGS[g], MGS	0.22	−0.30	$6.9e^{-14}$	$-3.3e^{-10}$	0.997	0.417	0.166	−0.042
West Coast	AA0404, AA0408, AA0414, AA1003	TBF, MGS	0.22	−0.30	$2.5e^{-13}$	$-2.4e^{-10}$	0.998	0.091	0.036	−0.009
Canterbury	AA0801, AA1003	TGS[g], MGS	0.21	-0.0^{6}	$6.7e^{-12}$	$-9.8e^{-11}$	0.989	0.935	0.370	−0.092
Otago and Southland	AA0403, AA0410, AA0414, AA0801, AA1003	TBF, TGS[g], MGS	0.21	−0.13	$4.8e^{-12}$	$-1.6e^{-10}$	0.992	0.748	0.297	−0.074

[a] AA0403—Fiordland temperate forests, AA0404—Nelson Coast temperate forests, AA0405—North island temperate forest, AA0406—Northland temperate kauri forest, AA0408—Richmond temperate forests, AA0410—Southland temperate forests, AA0414—Westland temperate forests, AA0801—Cantebury-Otago tussock grasslands, AA1003—South Island montane grasslands

[b] TBF—Temperate broadleaf and mixed forest; TGS—Temperate grasslands, savannas, shrublands; MGS—Montane grasslands and shrublands

[c] Characterization factors from de Baan et al. (2013a). Where more than one biome is included, a weighted average based on area is used. occ_b is used to show that these are CFs on biome level

[d] Adjusted CFs with pasture as reference

[e] Characterization factors from de Baan et al. (2013b). Where more than one ecoregion is included, a weighted average based on area is used. occ_e is used to show that these are CFs on ecoregion level

[f] Factors in accordance with Michelsen (2008). Where more than one ecoregion is included, a weighted average based on area is used. ΔQ is for native forest as reference, ΔQ^* for pasture as reference

[g] de Baan et al. (2013a) provide no CF for forest, used for TGS. Here world average is used, see text for details

When pasture is used as reference vegetation the results following Coelho and Michelsen (2014) and de Baan et al. (2013b) are correlated (correlation coefficient −0.885, p= 0.002), but not the others. This does not change if a worst case is used as CF for 'Forest, used' for the biome 'Temperate grasslands, savannas, shrublands'.

The variability of the methods is significant. Table 5 shows average values, standard deviation and the relative size of the

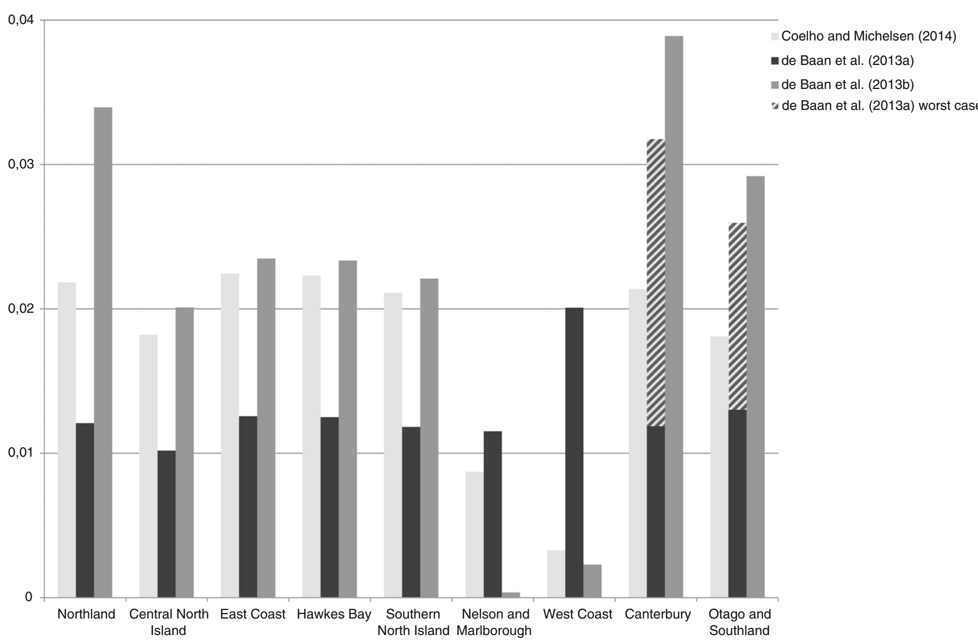

Fig. 3 The calculated biodiversity scores of *P. radiata* production for the different regions of New Zealand, using native vegetation as reference. The scores based on Coelho and Michelsen (2014) and de Baan et al. (2013a) are on the same scale, while the scores from de Baan et al. (2013b) are multiplied with 10^{11}. For Canterbury also a worst case using de Baan et al. (2013a) is included (*shaded area*)

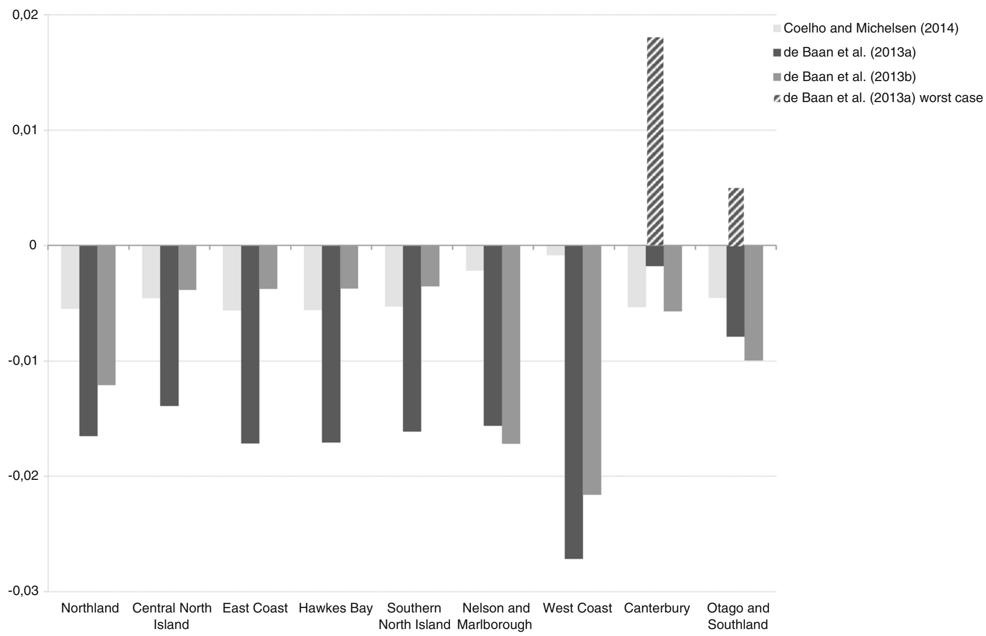

Fig. 4 The calculated biodiversity scores of *P. radiata* production for the different regions of New Zealand, using pasture as reference. The scores based on Coelho and Michelsen (2014) and de Baan et al. (2013a) are on the same scale, while the scores from de Baan et al. (2013b) are multiplied with 10^9. For Canterbury also a worst case using de Baan et al. (2013a) is included (*shaded area*)

smallest score compared to the largest score to show how the methods differentiate between the different areas.

When a worst case is used as CF for 'Forest, used' for the biome 'Temperate grasslands, savannas, shrublands', the average following de Baan et al. (2013a) increases to 0.017 (SD ±0.007) and the difference between the highest and lowest score increases to 3.1 when PNV is used as reference. Similar, the average increases to −0.011 (SD ±0.014) when pasture is used as reference land use.

4 Discussion

The question on how to include land use and land use changes in LCA and LCIA has been discussed in research literature for decades (Lindeijer 2000; Milà i Canals 2007a; Bare 2010; Koellner et al. 2013b). UNEP-SETAC recently published a guideline for the assessment of land use which can be used globally (Koellner et al. 2013b) and here there is a call for more consistency in methods. In this study, three different methods to assess impacts on biodiversity from land use are applied in order to assess the total impacts from forestry plantation in New Zealand as well as to investigate the correlation between the methods. Overall, we found substantial variation between regions and a significant divergence between the included methods.

The selected functional unit was 1 m^3 of wood, only including forestry activities. This selection of system boundaries and functional unit is often used for wood production since it is then not restricted to a specific use of the wood and

Table 3 Total impact assessed with the three selected methods with two different reference situations: native vegetation and pastures (*)

		Reference: native vegetation			Reference: pasture		
Wood supply region	ha×year	Coelho and Michelsen (2014)	de Baan et al. (2013a)	de Baan et al. (2013b)	Coelho and Michelsen (2014)	de Baan et al. (2013a)	de Baan et al. (2013b)
Northland	0.055	0.022	0.012	$3.4e^{-13}$	$-5.5e^{-3}$	−0.016	$-1.2e^{-11}$
Central North Island	0.046	0.018	0.010	$2.0e^{-13}$	$-4.6e^{-3}$	−0.014	$-3.8e^{-12}$
East Coast	0.057	0.022	0.013	$2.3e^{-13}$	$-5.6e^{-3}$	−0.017	$-3.7e^{-12}$
Hawkes Bay	0.057	0.022	0.013	$2.3e^{-13}$	$-5.6e^{-3}$	−0.017	$-3.7e^{-12}$
Southern North Island	0.054	0.021	0.012	$2.2e^{-13}$	$-5.3e^{-3}$	−0.016	$-3.5e^{-12}$
Nelson and Marlborough	0.052	0.009	0.012^6	$3.6e^{-15}$	$-2.2e^{-3}$	−0.016	$-1.7e^{-11}$
West Coast	0.091	0.003	0.020	$2.3e^{-14}$	$-8.2e^{-4}$	−0.027	$-2.2e^{-11}$
Canterbury	0.058	0.021	0.012	$3.9e^{-13}$	$-5.3e^{-3}$	−0.002	$-5.7e^{-12}$
Otago and Southland	0.061	0.018	0.013	$2.9e^{-13}$	$-4.5e^{-3}$	−0.008	$-9.9e^{-12}$

Table 4 Correlation of results obtained by the three methods from Coelho and Michelsen (2014) and de Baan et al. (2013a, b) based on Pearson correlation test

		Coelho and Michelsen (2014)	de Baan et al. (2013a)	de Baan et al. (2013b)
Coelho and Michelsen (2014)	Corr.coefficient	1.000	−0.688*	0.853**
	Sign. (2-tailed)	–	0.040	0.003
de Baan et al. (2013a)	Corr.coefficient	−0.688*	1.000	−0.453
	Sign. (2-tailed)	0.040	–	0.220
de Baan et al. (2013b)	Corr.coefficient	0.853**	−0.453	1.000
	Sign. (2-tailed)	0.003	0.220	–

Correlations at a 0.05 level are flagged * and correlations at a 0.01 level are flagged **. Native vegetation is used as reference

the results can be used as part of a range of product systems, such as construction materials, bioenergy, etc. (Berg and Lindholm 2005; Michelsen et al. 2008).

4.1 Variation between methods

The results based on Coelho and Michelsen (2014) are correlated with the results from the method described in de Baan et al. (2013b). There is also a weak correlation between results based on Coelho and Michelsen (2014) and de Baan et al. (2013a), but as described above this is dependent on choices made for missing CFs in de Baan et al. (2013a). The overall picture from the results in this study is consequently that the variation between the methods is remarkable.

It is however important to have in mind what the different methods actually measure. The method suggested by Michelsen (2008) and adapted for more general use by Coelho and Michelsen (2014) focuses on rareness and what primarily differentiates the WSRs is the values for ecosystem vulnerability (Table 2). Almost all plantations in the West Coast region are located in the ecoregion AA0404 Nelson Coast temperate forests and AA0414 Westland temperate forests, both regarded to be in a relatively stable/intact ecological condition (WWF 2013). The productivity in the West Coast region is low (Table 1) and the area requirement thus high, but the high area requirement is more than compensated by the low impact per area unit using this method. East Coast is identified as the WSR with the highest impact. There are only small differences compared to, e.g. Northland and Hawkes Bay. All these have values on ecosystem vulnerability equal to 1.

The characterization factors for the different WSRs are almost equal using the method proposed by de Baan et al (2013a), with the exception of Canterbury and Otago and Southland if a worst case scenario is used (Table 2). Since the CFs do not cause significant differences for the WSRs, the differences in impact for the WSRs are following this method caused by different area requirement, identifying the West Coast as the area with the highest impact. This changes if a worst case is calculated since the CFs for Canterbury and Otago and Southland changes significantly. The result is that these two WSRs have the highest impact, despite having lower area requirements than the West Coast (Fig. 3).

The method proposed by de Baan et al. (2013b) differentiates the WSRs most (Table 5). In a similar manner as the results following Coelho and Michelsen (2014), it is the CFs and not the area requirement that dominate the results, and also here, the West Coast region with the highest area requirement still has the lowest impact due to the low CF for the region. Further examining the CFs presented in Table 2, there is only three ecoregions that are given a CF different from zero by de Baan et al. (2013b): AA0405, AA0406 and AA1003. The characterization factor given in Table 2 is consequently depending on the fraction of the WSRs found in these ecoregions and even if only 7 % of the plantations in Canterbury are found in ecoregion AA1003, these plantations are the reason why this WSR gets the highest CF. The WSR Northland has the second highest CF. Here, the entire WSR is found within AA0406, but the CF for AA0406 is only 6 % of the CF for AA1003 (de Baan et al. 2013b) and land use in the WSR Northland still gets a lower overall impact than in Canterbury.

Table 5 Average scores, standard deviation and ration between maximum and minimum value for the three selected methods with two different reference situations; native vegetation and pastures

	Reference: native vegetation			Reference: pasture		
	Coelho and Michelsen (2014)	de Baan et al. (2013a)	de Baan et al. (2013b)	Coelho and Michelsen (2014)	de Baan et al. (2013a)	de Baan et al. (2013b)
Average score	0.017	0.013	$2.2e^{-13}$	−0.004	−0.015	$-9.1e^{-12}$
SD	±0.007	±0.003	$\pm 3.9e^{-13}$	0.002	0.007	$6.7e^{-12}$
max/min	6.8	2.0	107.1	6.8[a]	15.4[a]	6.1[a]

[a] here min/max is used

When pasture is used as reference vegetation, all methods show an environmental improvement for all WSRs with the exception of Canterbury and Otago and Southland when a worst case scenario is applied for the method from de Baan et al. (2013a). To some degree, the results present the same trend as when PNV is used as reference (Fig. 4). Following Coelho and Michelsen (2014), West Coast still give the least changes due to a very small change in assessed quality, as a strong contrast to both methods from de Baan et al. (2013a, b) that identify West Coast as the area where a changed land use from pasture to plantation forest gives the largest improvement due to the large area requirement. Since the environmental change is regarded as positive, large areas (due to low productivity) will in general show a larger improvement.

Following de Baan et al. (2013a), Canterbury is the area with least improvement, or even an overall increased environmental impact. This is due to the low negative (or positive for the worst case scenario) characterization factor, not the area requirement. The areas with the least environmental improvement following de Baan et al. (2013b) are caused by a combination in small changes in quality (low characterization factors, cf. Table 2) and low area requirement.

4.2 Comparison to other studies

Michelsen's (2008) method was originally developed and applied on a case study in boreal forests. Michelsen (2008) found values for production of Norway spruce (*Picea abies*) ranging from 0.102 to 0.186 per m^3 timber. These scores can be recalculated based on the proposal from Coelho and Michelsen (2014) using hemeroby values from Brentrup et al. (2002). Naturalness class H4 (e.g. managed forests with unnatural high share of conifers) in Brentrup et al. (2002) should then be used, changing CMB to 0.6 and reducing the overall impact to 0.093–0.133 per m^3 timber in the Norwegian case. In any of these situations, the impact from land use on biodiversity are substantially higher for the Norwegian case in Michelsen (2008) than the scores for *P. radiata* plantations found in this study (0.003–0.022).

Even though the methods proposed by de Baan et al. (2013a, b) to our knowledge are not applied on other forestry cases than the one presented here, at least the method proposed by de Baan et al. (2013a) can be applied on the case described in Michelsen (2008). In the Norwegian case study, forestry was present in two ecoregions in two different biomes: PA0520 'Scandinavian costal conifer forest' which is part of the biome 'Temperate coniferous forest', and PA0608 'Scandinavian and Russian taiga' which is part of the in the biome 'Boreal forest/taiga'. de Baan et al. (2013a) do not provide a characterization factors for 'Boreal forest/taiga', but the characterization factor for 'Forest, used' in 'Temperate coniferous forest' is 0.15, giving a total score on 0.050 for the case in PA0520 in Michelsen (2008). For comparison, the worst case from New Zealand following de Baan et al. (2013a) have a score on 0.020 (Table 3). de Baan et al. (2013b) do not provide characterization factor for any of the ecoregions used in Michelsen (2008) and this method can consequently not be used on this case.

4.3 Potential alternative methods

There are other proposed methods for the inclusion of land use impacts on biodiversity in LCA. Our view is that they are too generic and consequently unable to differentiate the areas of interest at the level required in this study. Peter et al. (1998) developed a forestry-specific method, but this method was not found relevant for our case study. Some of the indicators are too subjective since 'appropriate measures' are requested on a five graded scale and a clear definition on what is meant by 'appropriate' is not given. Most important was still the fact that all plantations most likely would score exactly the same and the method would consequently not give any valuable information on differentiating the WSRs. The same problem occurs if the Biotope method developed by Kyläkorpi et al. (2005) is applied. If PNV is used as reference, all plantations changes from a 'critical' to a 'general' biotope and the only difference will be area requirement, i.e. productivity. If pasture is used as reference, there will be no changes and the impact will be regarded 0 for all WSRs.

de Souza et al. (2013) have also developed characterization factors for different land use on a biome level based on functional diversity. This could have been an interesting alternative to the three included methods, but they do not provide characterization factors for any of the relevant biomes in this study and it is consequently not possible to test the method on this case.

4.4 Geographic dependency

The results in this case study show that geographic location of activities has a large effect on the valuation of impacts on biodiversity. This is of course a central assumption for developing methods with geographic dependency, but it might seem like some gradients are unrealistically steep, even for the more fine-graded methods. As mentioned, following the method from Michelsen (2008), the most significant factor is the ecosystem vulnerability and this can change from 1 to under 0.1 for neighbouring areas (Coelho and Michelsen 2014). Coelho and Michelsen (2014) showed that when New Zealand-specific data was applied instead of the global datasets from WWF (2013), in particular values for ecosystem vulnerability seemed to level out and the steep gradients were no longer present. This illustrates the trade-off between accurate methods and feasible data requirements.

The total scores based on the method from de Baan et al. (2013b) are dependent on the share of the plantations in a

WSR found in ecoregions with a characterization factor different from zero. Of the 10 ecoregions relevant, only three have a CF different from 0, which basically means that a change from native forest to plantations of *P. radiata* in the other seven ecoregions is regarded to have no effect on biodiversity. Whilst plantation forestry does offer some biodiversity benefits, this assessment is possibly an underestimate and should be a topic for more detailed investigations.

The actual location of the plantation is in itself an important factor due to the geographic dependency. This supports the need for spatial data for the activity under study. Here, we have used the relative share of plantations in a WSR within each relevant ecoregion for calculating new characterization factors. This is important for the final results. As an example, in the WSR West Coast, more than 47 % of the area is within ecoregion AA1003 South Island montane grasslands, but less than 0.3 % of the WSR's plantations are found within that ecoregion. Such accurate assessments were made possible by the use of GIS and in the absence of such tools erroneous estimates could compromise the final results.

4.5 Research priorities

The case study reveal that the results from the included methods give different results and recommendations drawn on the methods included would consequently go in somewhat different directions. As already mentioned, this is not any surprise since the methods focus on different aspects of biodiversity but it clearly identifies the discrepancies in present methods on impacts from land use. Here, it is also important to have in mind that these methods are all limited to impact from land occupation as driver for impact on biodiversity. Several other factors not included in either of the methods might also have a significant impact: land transformation, background climatic conditions, landscape fragmentation and turnover, introduction of new species and soil depletion amongst others. Also, PNV is here set equal to native vegetation. Even this is suggested by Koellner et al. (2013b), this is debated (Chiarucci et al. 2010). More detailed studies on reference situation are thus needed.

A larger number of locations and cases could have given different results on the correlation between the methods; here nine WSRs regions and one land use impact (*P. radiata* plantations) are included, but this would not change the fact that as long as these methods have the main emphasis on different aspects of biodiversity, the results will necessarily vary. Nevertheless, more case studies where several methods are applied are required to inform, improve and mature biodiversity impact assessment in LCA.

In this study, only occupational impacts were included. Michelsen et al. (2012) concluded that transformation impact was of less concern for forestry if a high number of rotations are included, but transformation should still be taken into account if the number of rotations is unknown. This will often be the case when new plantations are established. It is at present also a problem that methods for transformation impacts are less developed than occupational impact.

At present, there is substantial scope in all impact assessment methodologies for an improvement in the underlying data and criterion. Similarly, the assessments of land use impact on biodiversity in life cycle inventories relies heavily on data, consequently compiling suitable life cycle inventory data is a research priority. Here, progress is done over the last few years; de Baan et al. (2013a, b) and de Souza et al. (2013) all provide datasets for at least some land use activities globally, and Coelho and Michelsen (2014) provide a method using already available datasets with global coverage.

5 Conclusions and recommendations

Despite considerable gaps in knowledge, the human impact on global biodiversity is irrefutable and steps to reduce human impact on biodiversity require immediate action. This study provides a first attempt to quantify land use impacts on biodiversity within a LCA platform across plantation *P. radiata* forestry production in New Zealand. Three different approaches are used and these give somewhat diverging results. In addition, there are limitations in the present biodiversity indicators so the presented data should be interpreted with caution.

The results in this study highlight the need for more research in this area as well as raising the question for more harmonized approaches. When different approaches give rise for different recommendations, the usefulness for decision-making will be questioned. However, different approaches answer different questions, and from our point of view, it is clear that developers of decision support tools, as LCA, must be explicit on what is included and emphasized in their methods, and what is not. Impacts on biodiversity from land use are of no difference; is e.g. rarity or absolute species diversity given priority?

The complexity of biodiversity is large and the approaches included emphasize different aspects: rarity, absolute and relative species richness. It is therefore expected that the results differ. It is our view that studies aiming to provide empirical site-dependent assessment, including additional region- or site-specific data, and those that relate management practices to impact, should be a priority. Furthermore, the indicators that we have trialed in this study focus on land use as the driver behind biodiversity assessment, but other drivers may also be influential and therefore future biodiversity methodologies will need to accommodate these factors (Curran et al. 2011). To achieve this for a credible and useable international methodology presents a number of difficult research challenges.

LCA is shown to be a useful tool for overall environmental assessments of products and services, but in order to fully implement impacts also from land use there needs to be parity and validity in the underlying methodology. Several authors have emphasized the role of appropriate case studies in maturing LCA concepts and this study has made a contribution to the evolution of biodiversity metrics. Given the complexity of the issue, the development of biodiversity methodology will be slightly different because a significant concerted effort drawing from hitherto unrelated disciplines is necessary. Whilst uncertainty inherent in all natural systems creates challenges to the accuracy and application of these mechanisms, the process and ensuing results are an equally important outcome.

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